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Title: Prioritisation approaches for substances of emerging concern in groundwater: a critical review

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Abstract

Risks from emerging contaminants (ECs) in groundwater to human health and aquatic ecology remain difficult to quantify. The number of ECs potentially found in groundwater presents challenges for regulators and water managers regarding selection for monitoring. This study is the first systematic review of prioritisation approaches for selecting ECs that may pose a risk in groundwater. Online databases were searched for prioritisation approaches relating to ECs in the aquatic environment using standardised key word search combinations. From a total of 672, studies 33 met the eligibility criteria, based primarily on the relevance to prioritising ECs in groundwater. The review revealed the lack of a groundwater specific contaminant prioritisation methodology in spite of widely recognised differences between groundwater and surface water environments in regards to pathways to receptors. The findings highlight a lack of adequate evaluation of methodologies for predicting the likelihood of an EC entering groundwater and highlights knowledge gaps regarding the occurrence and fate of ECs in this environment. The review concludes with a proposal for a prioritisation framework for ECs in groundwater monitoring which enables priority lists to be updated as new information becomes available for substances regarding usage, physico-chemical properties and hazards.

TOC/Abstract Art.

Keywords

Emerging contaminants; Groundwater; Prioritisation; Monitoring

1 Introduction

Research on substances of emerging concern in the aquatic environment has expanded in recent years. They are often referred to as 'emerging contaminants' (ECs) and as substances "*that are currently not included in routine monitoring programmes*" and "*may be candidates for future regulation, depending on research on their ecotoxicity"* and *"monitoring data regarding their occurrence*" in the environment.[1] In some cases they are also substances which still require the development of conceptual models to describe their behaviour and occurrence in the environment.[2] ECs include pharmaceuticals and personal care products (PPCP), illicit drugs, hormones and steroids, industrial substances, disinfection by-products and pesticide degradation products.[2-4] Approximately 860 ECs in the environment that are currently being researched or discussed.[1] There has been an increase in the monitoring of ECs in the environment, largely due to advances in analytical chemistry techniques. Contaminants can be detected in concentration ranges below 1 ng/l that were previously below the Limit of Detection (LOD).[3,5] New techniques include multi-residue gas and liquid chromatography techniques coupled with mass spectrometry.[3,5]

The potential risks from ECs to human health and aquatic ecology in the environment have been recognised,[6-8] and new standards and regulations may be required.[2,9-11] ECs are now understood to be "*ubiquitous contaminants in the environment*" and there is evidence that these contaminants can have disruptive effects to organisms at different trophic levels, including humans.[12] There is also growing concerns regarding the occurrence of pharmaceuticals in the environment and the build-up of antifungal and antibiotic resistance.[13]

There remain many challenges for regulators and water managers regarding the monitoring of ECs in the aquatic environment.[14] These challenges specifically relate to the lack of knowledge on their occurrence and fate, the number of ECs potentially present in the environment and the fact that many of them are unregulated.[14,15] This is a particular concern for groundwater because the environmental fate of ECs is still not well understood.[16,17] Groundwater is a valuable resource and amounts to 98% of the Earths' freshwater[18] and supplies approximately 50% of all drinking water globally.[19] Drinking water treatment might only involve disinfection, meaning there is a risk of ECs contaminating supplies from groundwater.[20] ECs have been detected in treated drinking water.[16,20] Groundwater is also vital to the health of groundwater-dependent ecosystems such as rivers, lakes and wetlands.[21]

1.1 Regulatory background

In Europe, the Water Framework Directive (WFD) (2000/60/EC)[22] requires Member States (MS) to manage water in an integrated ecosystem-based approach, and considers that all waters and their dependent ecosystems are inter-linked and inter-dependent. The key objective of the WFD is to establish good ecological status in all surface waters and good chemical and quantitative status in all groundwaters through a formal process until 2027. The WFD does not allow for deterioration in water body status.

The Groundwater Daughter Directive (GWDD) (2006/118/EC)[21] further describes how the chemical status of groundwater bodies is defined using Threshold Values (TVs). They indicate environmental risk and trigger the requirement for further investigation.[23] Many of the TVs relate to the protection of groundwater receptors such as rivers, groundwater-dependent terrestrial ecosystems or drinking water supplies.[16,17] For many 'classical' contaminants there is sufficient information about the pathways and toxicity to receptors; however, not enough is known about ECs to define TVs.[16]

A 'chicken or egg dilemma' prevails, as the gaps in knowledge relating to the occurrence and risk of ECs delay regulation and the lack of regulation delays the generation of monitoring data.[24] The number of ECs makes it difficult to identify which ones should be monitored.[25] In Europe, this has been addressed by developing a watch list for pollutants which aims "*to increase the availability of monitoring data on substances posing a risk or potential risk to bodies of groundwater, and thereby facilitate the identification of substances, including emerging pollutants, for which groundwater quality standards or TVs should be set*" (2014/80/EU).[26] The first watch list under the Priority Substances Directive (2008/105/EC[27] as amended by 2013/39/EU), has already been adopted for surface water in 2015. Ten new substances including 17α-ethinylestradiol,17β-estradiol and diclofenac are listed.[28]

A similar situation occurs elsewhere in the world for the regulation of ECs. The US Environment Protection Agency (USEPA) published a Contaminant Candidate List (CCL) for drinking water.[29] This is required under the Safe Drinking Water Act (SDWA) for contaminants known or anticipated to occur in drinking waters and may require regulation in the future.[29] The latest CCL (no. 4) from 2016 includes 97 chemicals from industrial use, pesticides, disinfection by-products and pharmaceuticals.

1.2 Emerging contaminants in groundwater

Most research on ECs in the environment focuses on wastewater and surface water, while there has been less emphasis on groundwater.[30] ECs have the potential to leach through subsoils to groundwater and have been detected in aquifers since the 1990s.[31] ECs may get into groundwater from numerous origins as shown in Figure 1, but wastewater has been identified as the primary source.[16] Point sources include private wastewater treatment systems, animal waste lagoons and landfill leachate.[16] Managed artificial recharge of partially treated wastewater or surface water (i.e. bank infiltration) can also be important sources of ECs in groundwater.[17] Diffuse sources include application of manure, pesticides, biosolids from sewage sludge, and atmospheric deposition.[16,17,32,33]

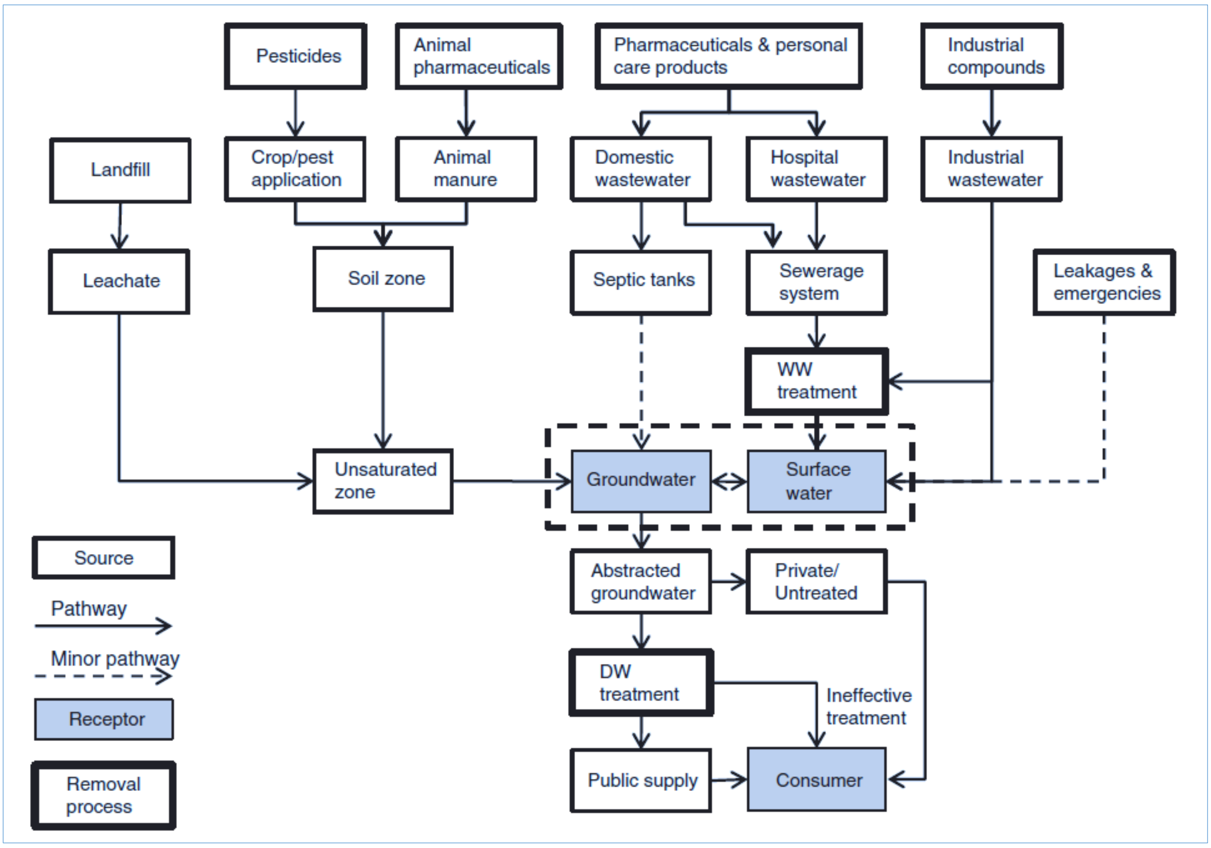


Figure 1 Sources of emerging contaminants and pathways towards receptors[17]

Numerous studies in the USA[34,35] and Europe[17,30,36] provide an overview of the occurrence of ECs in groundwater. A global review of studies[16] published since 1993 documented significant concentrations (10^2 to 10^4 ng/l) of ECs, which included a range of PPCPs (e.g. carbamazepine and ibuprofen), industrial compounds, and caffeine. Transformation products can be found more frequently, and in higher concentrations, than their parent compounds.[4,16]

Previous studies have demonstrated that concentrations of ECs in surface waters are higher than those in groundwaters.[7,34,35,37] In addition, the lists of ECs most frequently detected in groundwater differ from those in surface waters.[7,34,35,37] For example, a comparative survey[37] of 70 groundwater and 71 surface water samples in France, found that several pharmaceuticals detected in surface water were not present in groundwater. This is because the main source of ECs in the aquatic environment is wastewater effluent, which discharges directly into surface waters, while groundwater is generally less vulnerable to contaminants due to the protective properties of soils and the unsaturated zone. However, groundwater bodies in areas with an absence or only a thin layer of subsoils have increased vulnerability to contamination, including by ECs.[38,39] The occurrence of ECs in United Kingdom, French and Italian groundwater, also showed higher concentrations in karstic aquifers relating to high transmissivity, and conduits.[39,40] In addition to infiltration through the subsurface environment, another pathway of ECs to groundwater is via surface water-groundwater exchange.[16] There remain gaps in the understanding of EC sources, the pathways to receptors and toxicity mechanisms and levels.[2]

1.3 Prioritisation approaches for monitoring contaminants in groundwater

Given the lack of knowledge about the behaviour and impacts of ECs on groundwater receptors, many ECs are not routinely monitored in groundwater.[2] Both the number of ECs, and the fact that not all of them will be harmful to human health or the aquatic environment, means that prioritisation is required to develop cost effective monitoring programmes that target the highest risk ECs, which may warrant regulation in the future.[2,11,14] As demonstrated by previous studies[35,37], EC occurrence in groundwater can differ from surface water; in regard to the types of contaminants, detection frequencies, and concentrations. Consequently, it appears inappropriate to use priority lists developed for surface waters.

Existing techniques for prioritising chemicals are generally based on the principles of risk assessment.[25]

The risk is the probability of the occurrence of exposure of a chemical to a biological receptor multiplied by the associated effect, known as the hazard.[25] The way exposure and hazard are combined to calculate the risk, varies between prioritisation approaches and this can affect the results.[41] There is no standard approach for prioritising ECs in groundwater. The Common Implementation Strategy Working Group for Groundwater (CIS WGGW, 2018) has outlined a process for developing a voluntary groundwater watch list (GWWL) at an EU level.[42,43] The NORMAN Network[1], a group of stakeholders interested in emerging contaminants (which includes academia, industry and regulators), are also developing a prioritisation methodology for groundwater (currently unavailable).

Exposure relates to the environmental occurrence of a substance, which can be estimated using simple equations or environmental fate models.[40,44,45] Occurrence in groundwater is not solely dependent on source factors and the characteristics of the pathway also warrant consideration.[11,16] Migration through the subsurface is determined by several factors,[4,46-49] such as physico-chemical properties of the compounds as well as those of soils and subsoils. Indices have been developed for estimating the leaching potential of contaminants (mainly for pesticides).[50] Existing prioritisation approaches for groundwater have used these for characterising environmental exposure. For example: the Groundwater Ubiquity Score (GUS index)[51] based on the physico-chemical properties of the compounds was used to prioritise pesticides in South African groundwater[52]; and the Attenuation Factor (AF) also based on the physico-chemical properties, as well as soil properties, the subsurface depth and recharge[53] was used for estimating the leaching quantity of sixteen ECs in Ireland[54].

While there are numerous studies on the prioritisation of chemicals in the aquatic environment, there is a lack of consensus on critical components, such as determining exposure in groundwater and quantifying the hazard.[10,55] Only one published study[2] so far had specifically set out to review prioritisation for groundwater monitoring, but neither analysed approaches in detail nor proposed any groundwater specific techniques. Consequently, there is a need to review prioritisation techniques to determine the best approach for prioritising ECs in groundwater. This will help to focus groundwater monitoring efforts on those ECs that present the highest risk to human health or ecological receptors.

To the authors knowledge this paper provides the first critical review of prioritisation approaches for selecting ECs for monitoring that may pose a risk in groundwater. It reviews existing approaches to provide a synthesis of their elements which may be appropriate for groundwater and to identify knowledge gaps. The specific objectives of the review are to: 1) review existing prioritisation approaches for ECs with an emphasis on methodologies that can be used for groundwater; 2) evaluate the methodologies within these prioritisation approaches for predicting EC occurrence in groundwater; 3) analyse the prioritisation results from a subset of studies to examine similarities and differences, and the impact of an approach on the result; and 4) describe a framework for a prioritisation approach for ECs in groundwater and make recommendations for further research.

2 Systematic review criteria

The systematic review was conducted following the general principles published in "*The Production of Quick Scoping Reviews and Rapid Evidence Assessments*".[56] A predefined protocol was developed by the authors and extracts are available in Supplementary Information A. The keywords searched are outlined Supplementary Information A, the record of results returned is in Supplementary Information B. The search source of published literature was the online database Scopus. Some of the recent work in this field has not been published within peer-reviewed journals, therefore, websites of relevant specialist organisations such as USEPA and EU Joint Research Council (JRC) were also searched.

Following the screening of titles and abstracts, the remaining articles were examined in full to determine their eligibility for inclusion in further assessment. This selection generated a more focused group of studies to improve quality and the confidence of the analysis relating to the research question. A predefined scoring system for relevance and quality was developed as part of the protocol (Supplementary Information A). Studies were included if they had a relevant outcome (i.e. a prioritised list of chemicals for water quality monitoring purposes) and were of sufficient quality. The quality was determined through a process of critical appraisal to ensure only reliable studies were included. For example, criteria included: the study having a clear aim and transparent methodology. For the eligible studies, key information on the prioritisation approaches was extracted (see Supplementary Information A).

3 Prioritisation approaches for monitoring ECs in groundwater

3.1 Study characteristics from systematic review

A total of 672 studies were identified and following the screening and eligibility assessment 33 studies were deemed eligible for inclusion in the review. The primary reason for exclusion was the lack of a relevant outcome, that is, the study did not include a prioritised list of chemicals for water quality monitoring purposes. Summary results from this systematic review of published prioritisation studies are shown in Figure 2 (see Supplementary Information C (Table D) for details of the study characteristics). All studies were available in the English language and dated from 2003 to 2016. Many were published since 2014, accounting for 50 % of the studies included.

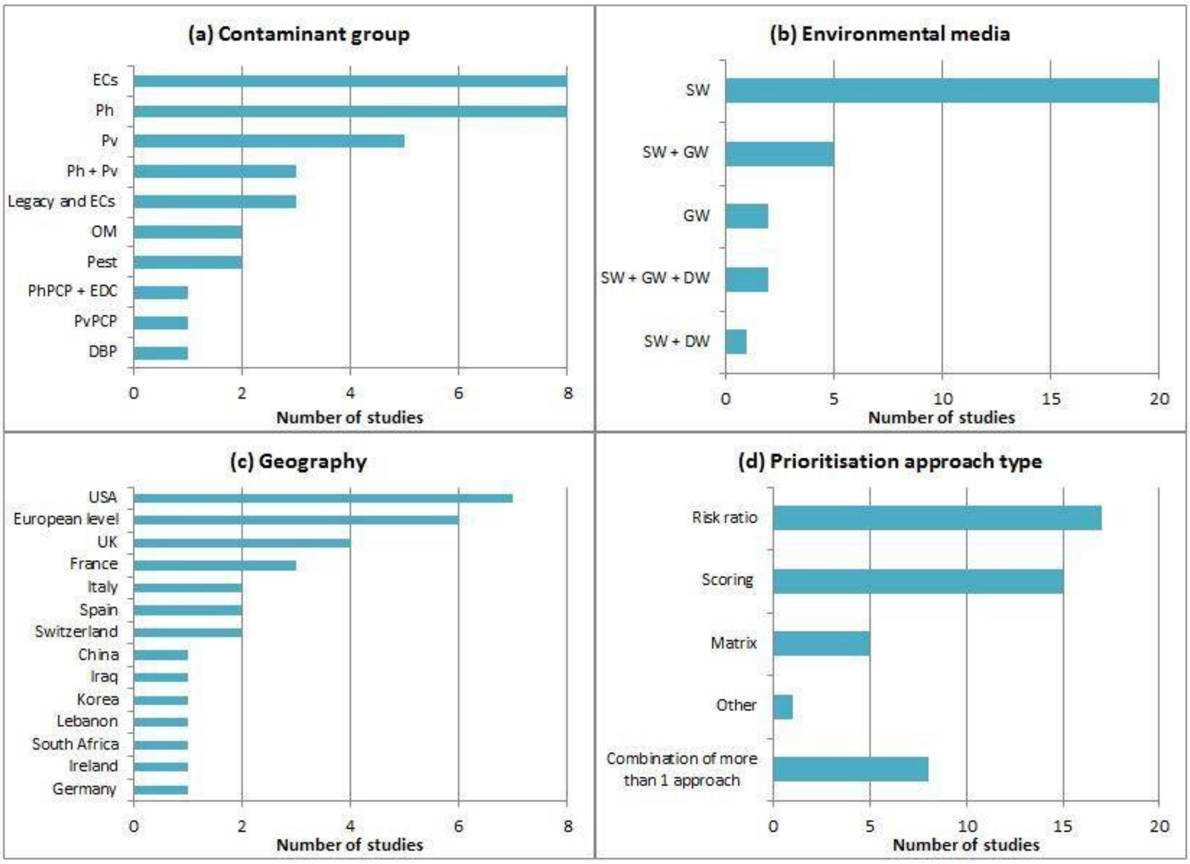


Figure 2. Summary results from review of studies of prioritisation approaches for ECs, a) key contaminant groups included in published prioritisation approaches, b) range of environmental media, c) geographical coverage of studies, d) types of approaches for prioritisation. SW=surface water, GW=groundwater, DW=drinking water. Ph=human pharmaceuticals, Pv=veterinary pharmaceuticals, Legacy=regulated legacy contaminants, Pest=pesticides, OM=organic micropollutants, PCP=personal care products, EDC=endocrine disrupting compounds, DBP=disinfection by-products.

Eight of the studies focused on ECs as a general category, with a further three examining legacy contaminants and ECs together (Figure 2a). Two studies were aimed specifically at organic micropollutants, three related to pesticides and one to disinfection by-products. Half of the studies (n = 17) focused on pharmaceuticals in the environment. Ten of these studies were for human pharmaceuticals, five for veterinary and two for both. Two of the studies focusing on pharmaceuticals also included personal care products.

Only a very small proportion of published prioritisation studies were found to have a groundwater focus (Figure 2b). The majority of the studies were aimed at surface water (n = 21). A total of nine studies related to groundwater, with only two focused on groundwater (see Supplementary Information C (Table E) for full table of results). Five studies were aimed at both surface water and groundwater, and a further two for surface water, groundwater and drinking water.

The selected studies were conducted in 13 different countries or regions (Figure 2c). Most of the studies were undertaken in European countries (n = 21), six at a European scale and four based in the United Kingdom. The USA also accounted for a significant number of the studies (n = 7). Five studies were from other countries (Korea, China, South Africa, Iraq and Lebanon).

The examined studies represent several different approaches of combining exposure and hazard assessments to determine risk, including the risk ratio approach, and scoring systems or matrices (Figure 2d). Seventeen studies followed the risk ratio approach which was used slightly more frequently than the scoring system approach. The risk ratio approach relies on having the dose-response toxicological data for the relevant trophic levels and receptors but is considered a simple to use method and it is easy to communicate the results.[57] A value above one indicates risk and may activate the substance's inclusion in monitoring programmes.[10,44,58] The scoring approaches involved categorising and combining scores for exposure and hazard. For example, for exposure leaching indicators can be used and for hazards, classification data can be used instead of dose-response data. There were 15 scoring system approaches, three of which also used a matrix approach for combining the scores. Six of the studies used a combination of the risk ratio and scoring system approach. Examples were the EU WFD prioritisation studies[58-60], where their first stage screening involved scoring chemicals based on the persistence, bioaccumulation and toxicity (PBT) approach and then the second stage prioritisation was based on the risk ratio approach.

Only a very small proportion of published studies were found to have a groundwater focus and of those only one covered ECs specifically (Figure 2). This highlights the limited attention that has been paid to groundwater and groundwater receptors to date. The number of substances and groups of substances covered so far for both surface water and groundwater is also very limited, and the geographical coverage biased to Europe and USA. This emphasises the need for prioritisation approaches to now look beyond traditional hotspots of surface water and wastewater systems and consider approaches that are appropriate for the protection of groundwater bodies. A wider geographical scope is needed, and risk to groundwater from ECs may be region or country specific in terms of substances used, quantities used, as well as pathways for potential groundwater contamination. The tendency for prioritisation of pharmaceuticals, could lead to some other ECs escaping scrutiny.[54] This may be referred to the "*Matthew effect*" whereby "*the prominence of a few contaminants targeted for investigation is dictated largely by the attention devoted to them in the past*". [54] Some of the prioritisation studies[10,57,61] which examined ECs and some classical contaminants had only included few pharmaceuticals in their priority lists.

3.2 Limitations of the study

This review has several limitations including a risk of bias in the results because there were repetitions in the prioritisation approaches included. About half of the studies only updated existing approaches or applied them, and sometimes the same authors were involved in more than one study. This can result in showing trends in the approaches used, just because they have been used previously. The same is also true for the types of ECs studied due to the focus on pharmaceuticals. It was beyond the scope of this review to consider unpublished prioritisation approaches and therefore other approaches for prioritisation of ECs in groundwater may be applied in some countries that were not included. As discussed in Section 6 there were limitations with comparing the results of different prioritisation approaches and these should be addressed in future to help verify the results.

4 Approaches for assessment of environmental exposure of ECs in groundwater

This section describes the trends in the methods for exposure assessment, their applicability to the groundwater environment and highlights the strengths and weaknesses. Table 1 provides an overview of the characteristics of the exposure (and hazard) assessment in each of the studies.

Table 1 Summary approach to exposure and hazard assessment

	Exposure									Hazard													
	Generic				Surface water		Groundwater			Receptors		Dose response					Classification						
Reference	Chemical Property	Sales	Usage	Metabolism	Predict conc SW	Measured conc SW	Predict conc GW	Measure conc GW	Leaching Indicators	Ecology	Human	Algae	Daphnia	Fish	Mammalian	Human dose	Persistence	Bioaccumulation	Carcinogenicity	Mutagenicity	Teratogenicity	Endocrine disruption	Neurotoxicity
Boxall *et al.* [62]			•	•						•		•	•	•									
Capleton *et al.* [63]		•		•							•					•		•	•	•	•	•	•
Besse and Garric[64]		•		•	•					•		•	•	•		•		•					
Kim *et al.*[65]		•		•						•	•					•							
Kools *et al.*[66]	•				•					•						•							
USEPA29,67		•	•			•		•			•					•			•	•	•		
Götz *et al.*[25]	•	•				•																	
Hebert *et al.* [68]						•		•			•								•	•			
Kumar and Xagoraraki[69]						•				•	•	•	•	•				•	•	•	•	•	•
Murray *et al.*[70]						•		•			•					•							
Daginnus *et al.*[59] (WFD)	•	•			•					•	•	•	•	•			•	•	•	•	•		
Diamond *et al.*[57]						•				•		•	•	•			•	•				•	
von der Ohe *et al.*[10]						•				•		•	•	•									
Coutu *et al.*[71]										•	•					•		•	•	•			
Sui *et al.*[11]		•			•					•		•	•	•				•					
Ortiz de García *et al.*[72]						•				•		•	•	•			•	•					
Bouissou- Schurtz *et al.*[73]	•				•	•				•		•	•	•									
Dabrowski *et al.*[52]	•	•	•						•		•								•	•	•	•	•
LaLone *et al.*[74]										•				•									
Maruya *et al.*[75]					•	•				•				•									
Carvalho *et al.*(JRC)[58]	•	•			•					•	•	•	•	•		•							

Reference	Exposure									Hazard													
	Generic				Surface water		Groundwater			Receptors		Dose response					Classification						
	Chemical Property	Sales	Usage	Metabolism	Predict conc SW	Measured conc SW	Predict conc GW	Measure conc GW	Leaching Indicators	Ecology	Human	Algae	Daphnia	Fish	Mammalian	Human dose	Persistence	Bioaccumulation	Carcinogenicity	Mutagenicity	Teratogenicity	Endocrine disruption	Neurotoxicity
(WFD)																							
Chirico *et al.*(JRC)[60] (WFD)	•	•			•					•	•	•	•	•			•	•	•	•	•	•	
Di Nica *et al.*[76]	•		•		•					•		•	•	•									
Ki *et al.*[50]	•		•						•														
Kuzmanović *et al.*[77]						•				•		•	•	•									
Al-Khazrajy and Boxall[78]			•		•					•	•	•	•	•		•							
Busch *et al.*[79]						•				•		•	•	•									
CIS WGGW[43]	•							•	•	•	•	•	•	•			•	•	•	•	•	•	
Clarke *et al.*[54]	•				•		•				•			•									
Donnachie *et al.*[61]						•				•		•	•	•									
Guo *et al.*[80]		•			•					•	•	•	•	•	•	•							
Mansour *et al.*[81]		•			•					•	•	•	•	•	•		•	•					
Sangion and Gramatica[82]										•		•	•	•									

Notes: a. Exposure assessment not included as part of this study. b. Hazard assessment not included as part of this study. Conc = concentration; SW = surface water; GW = groundwater.

The application of Measured Environmental Concentration (MECs) as a measure of environmental exposure was found to be a common approach (n = 17). MEC values from surface water were utilised in 13 of the included studies and four studies applied MEC values from groundwater. The use of MECs for groundwater are discussed further in Section 4.1.

Calculating Predicted Environmental Concentrations (PEC) was also a common approach with a total 14 studies using this approach to characterise exposure. Only one study calculated PECs specifically for groundwater. Sales or usage data was frequently used to estimate PECs (11 studies). It can be difficult to obtain the data required on sales, usage and environmental releases of ECs relevant to groundwater exposure. For example, it would be an enormous task to obtain usage data for all pharmaceutical compounds in the United Kingdom, and this type of information is not currently systematically reported or acessable.[80] Two studies did not use sales or usage data to calculate the PEC: one study[75] used wastewater effluent data to calculate PECs in surface water and the other study[66] on veterinary pharmaceuticals used estimates of the number of animals.

Five of the studies that calculated the PEC in surface water were for human pharmaceuticals using the European Medical Agency (EMA) guidelines[83]. Two of the studies calculated the PEC in surface water for veterinary pharmaceuticals using another EMA guideline[84]. The PEC in surface water was calculated using a European Centre for Ecotoxicology and Toxicology of Chemicals (ECETOC) Targeted Risk Assessment (TRA) Tool by the three WFD studies[58-60], the first in 2011[59] and then in 2015[58,60]. The latter also applied the FOCUS model in addition to the PEC calculation of pharmaceuticals in wastewater.[58,60] PECs were therefore calculated mostly for surface waters using established methods for specific contaminant groups such as human or veterinary pharmaceuticals, or pesticides. In addition, none of these studies used MECs to validate the PECs. Only one study[73] compared PEC and MECs and found a poor relationship that was not not scrutinised as part of the study.

Five studies used neither MEC or PEC for characterisation of exposure in the water environment. Three of the studies[62-64], used sales or usage data as an indicator of exposure in surface water. Each of the studies were for veterinary pharmaceuticals and all involved the same author which may bias the results.

Overall only four studies[43,50,52,54] predicted concentrations in groundwater or the likelihood of a contaminant entering groundwater (see Supplementary Information C). Two studies used neither the MEC or PEC approach and instead used leaching indicators: one study used the extended Attenuation Factor (AF)[50] and another used the Groundwater Ubiquity Score (GUS index)[52]. The Groundwater Watch List (GWWL) study[41] proposed a leaching indicator scoring system based on chemical properties and also incorporated MECs of ECs where they are available. The fourth study[54] used a model which incorporated the AF and an application rate for biosolids to calculate PEC. Two of the studies focused on pesticides[50,52] and two studies[43,54] covered ECs; one of which considered only one source (spreading of biosolids)[54] and the second[4342] considered all ECs but this methodology has yet to be implemented at the EU level and will be done on a voluntary basis.

The review highlighted that there is no trend in the methodologies for predicting concentrations of ECs in groundwater or the likelihood of a contaminant entering groundwater. Therefore, rationales and the limitations of these methodologies have been examined in further detail in Sections 4.1 and 4.2.

4.1 Measured Environmental Concentrations (MEC)

The application of MECs as a measure of environmental exposure has been demonstrated as a common approach. It is a reliable representation of environmental exposure because the results represent actual occurrence rather than estimates. However, there is a dependency on availability of monitoring data which there may be a lack of for ECs. The EU WFD surface water prioritisation studies[58-60] used monitoring data where possible and modelling was undertaken for substances where an insufficient quantity of monitoring data was available.[58] The USEPA[29,67] similarly used environmental release data and production data in the absence of MECs.

There are further considerations when the MEC approach is used, which include how to summarise the data to be representative of the risk, and dealing with results below the LOD. Studies that used MECs commonly incorporated both the frequency of detection of a compound, as well as the magnitude of its concentration. Frequency addresses regularity of occurrence and the magnitude addresses intensity. Concentrations could change over time which is difficult to capture[57] with the paucity of monitoring data of ECs in groundwater in particular. The magnitude can be represented by the mean concentration, the maximum or both.[29,43,67] Most studies opted for the conservative approach of using the maximum concentration (e.g. [10,70,73,77]). One study[10] calculated the 95-percentile concentration of the sites to help account for spatial variations. Only a few studies reported whether the MECs were influenced by point sources such as wastewater treatment plants.

The studies differed in their approaches to dealing with values below the LOD. Several studies[10,29,67] truncated the dataset by excluding data below the LOD. Alternatively, one study[71] left censored data by replacement with the highest LOD value to take a more conservative approach. Caution should be taken when dealing with MEC datasets with a high proportion of censored results, (i.e. < LOD) and substitution methods, such as replacing non-detects with half the detection limit or zero, are not recommended for calculation summary statistics (mean, median, quartiles). Statistical approaches such as Maximum Likelihood Estimation (MLE) and Regression on Order Statistics (ROS) should be used for estimating summary statistics.[84]

It can be argued that excluding less than values is appropriate in the context of prioritisation, because highly toxic chemicals that are frequently monitored but not often detected would result in a high risk, when in fact any risk is more likely to be low.[10] Conversely, disregarding MECs below the LOD could possibly lead to an underestimation of the real risk if the ECs are hazardous but present at low levels that could still be harmful to human health or the environment.

4.2 Exposure assessments for the likelihood of a contaminant to enter groundwater

4.2.1 Physico-chemical properties of ECs

The likelihood of a contaminant entering groundwater is considered higher if the contaminants' sorption coefficient is low (indicating higher mobility) and persistence is high,[86] where persistence is defined as the long-term exposure to an organism and is based on the half-life data. [87] Two studies[41,52] focussed on physico-chemical properties of ECs in their prioritisation.

The GWWL methodology[43] used a simple scoring system to indicate the likelihood of an EC reaching groundwater. The REACH guidelines[87] provide indicators for persistent chemicals based on the halflife in water >40 days (P) and >120 days indicating very persistent (vP) chemicals. The GWWL scoring methodology for persistence was consistent with this and the Pesticide Properties Database (PPDB).[86]

The GWWL methodology[43] proposed two indicators for mobility: logK_{oc} and logK_{ow}. K_{oc} is the ratio of the concentration of the contaminant that is sorbed to the organic carbon in the soil versus that which is in solution.[88] The K_{ow} relates to the equilibrium partitioning of a contaminant between octanol and water phases and is a surrogate for K_{oc}.[88] The GWWL methodology[43] gave higher risk scores for contaminants less likely to sorb to the soil and therefore more likely to reach groundwater. While this criterion is reasonable, the actual thresholds for the scoring in the methodology were neither explained nor justified.

The second study[52] to focus on physico-chemical properties of ECs utilised the GUS index as an indicator of environmental exposure to prioritise pesticides in South African groundwater. The GUS index applies the K_{oc} and the half-life in soil[52] and is widely used as an indicator of pesticide mobility[52,89]. They scored the pesticides with a GUS index of greater than 2.8 as highly mobile and those with a value of less than 1.8 as non-leachers. Again, this is consistent with the REACH guidelines[87] and the PPDB[86].

Using EC physico-chemical properties has merits as a screening tool for determining the likelihood of an EC entering groundwater relative to other substances. However, there are also some obvious drawbacks and uncertainties. It is difficult to predict the half-life and mobility of chemicals in environmental field conditions, and they are dependent on variables including temperature, pH, moisture, microbial populations and the soil type.[86,90,91] Many authors[92,93] have illustrated that chemicals can be neutral or ionic depending on the soil pH and therefore their sorption capacity changes, and as a result logK_{ow} may not the be most universally appropriate indicator.[94] Therefore, these methods are more appropriate for non-polar organic chemicals, such as pesticides, where they contribute to a better understanding of environmental fate and transport dynamics.[95] However they may not be appropriate for non-polar ECs such as pharmaceuticals. Other studies have demonstrated a relationship between the frequency of detection of pesticides and the GUS index, but also revealed that some presumed "non-leaching" pesticides were actually detected in groundwater.[91]

Neither study tested the sensitivity of their results for these leachability indicators. Also, the results were not verified by comparing the higher risk ECs with environmental data. These findings highlight that approaches that only use physico-chemical properties of ECs as leachability indicators can potentially mask or overestimate risks.

4.2.2 Pathway to groundwater

The vulnerability of groundwater to ECs is also dependent on many other factors including the physicochemical properties of the soil and subsoil, the depth to groundwater and the recharge rate. The AF (attenuation factor) is a simple index for ranking the leaching potential of pesticides and has been frequently used in the past (e.g. [95-97]). It was proposed in 1985[53] and is based on the half-life of the pesticide, depth of the soil, bulk density, organic carbon, sorption coefficient and recharge rate.

The extended AF was utilised in one study[50] as part of a geospatial leaching tool for agrochemicals in the USA. It accounted for the properties of Volatile Organic Compounds (VOCs) as well as pesticides, by adding in the dimensionless Henry's constant (Kh) (air partition coefficient) and the diffusion coefficient in soil. They used digital mapping of annual pesticide usage, soil properties and recharge to examine the variation in potential leaching loads over a regional scale, and found it could distinguish between areas of high and low susceptibility.

The second study[54] to apply the AF adapted a model for estimating PECs of pesticides in groundwater for sixteen organic ECs detected in biosolids in Ireland. It calculated the leaching quantity as a function of the AF and the application rate, the fraction intercepted by the crop and the thickness of the unsaturated zone.

Neither study attempted to verify their methods by comparing results with actual groundwater monitoring data. However, both studies[50,54] did undertake a sensitivity analysis on the parameters and found that K_{oc} and soil organic carbon were the most sensitive. In research into uncertainty analysis on the AF method it was found that a small variation in the retardation factor (i.e. retention in the mobile phase) could lead to different prioritisation classifications[98]. The authors of the study using the geospatial leaching tool[50] did acknowledge issues with the spatial and temporal map resolution. They emphasised the trade-off between the data availability and the accuracy of the predictions and concluded that their tool should only be used as a first step rapid and large-scale tool.[50] Approaches that incorporate geographical information at a regional scale are now common practice (e.g. [99-103]). Soil and groundwater models are considered to be less appropriate for generic risk assessments for determining monitoring programmes, as they can be too site specific.[44]

A better understanding of the fate and transport of ECs in groundwater is required to inform risk assessments, particularly their sorption and degradation.[16,104] In recent years, there have been several studies on the leaching potential of certain ECs, predominantly pharmaceuticals. For example, one research study[104] examined the irrigation of soil columns and irrigated fields to assess the leaching potential of acidic pharmaceuticals (ibuprofen, gemfibrozil, naproxen, ketoprofen, and diclofenac). At higher pH values (>8) these compounds tended to take their ionised more soluble state which increased their leaching abilities. However, no contamination of these pharmaceuticals in groundwater was observed.[104] Another study[105] found differing sorption of PPCPs with triclosan and octylphenol being moderately to strongly sorbed and negligible for carbamazepine. These authors demonstrated that microbial activity and soil organic carbon were important for the degradation. The relative persistence (28 to 39 days in unsterilized soils) and poor sorption of carbamazepine indicated that it is more likely to leach to groundwater.[105] Other studies have also highlighted carbamazepine as being relatively persistent and being prone to accumulate in soil.[104,106,107] Detections of carbamazepine have been observed in groundwater possibly as a result of the long-time available for downward migration due to its high persistence.[104] Consequently there are many factors that may determine the presence of ECs in groundwater and research into their persistence and sorption capabilities in environmental conditions is still on-going. In addition, the lag time between environmental releases and the potential occurrence in groundwater needs to be considered when attempting to verify prioritisation approaches.

4.3 Outlook for exposure assessments for the likelihood of an EC entering groundwater

The review found that the use of MECs is the preferred method for surface water and groundwater and more reliable more for representing environmental exposure especially for groundwaters where it is difficult to estimate the concentrations. Careful consideration is required when summarising data and dealing with results below the LOD so that the data is representative of the risk of exposure. Data should be summarised using statistically sound methods that are appropriate for the particular MEC dataset.

However, there is still insufficient monitoring data for most ECs and therefore estimates will still be required.[42] It is important therefore to generate a comprehensive list of ECs that have the potential to occur in groundwater that may not yet be measured. This initial list could be vast and therefore should be drawn up with the involvement of stakeholders.[59] There is still a dependence on the availability of sales and usage data and data on the physico-chemical properties of ECs which is not as accessible as other contaminant groups that are regulated such as pesticides.[42]

Unlike for surface water there are no standard methods for calculating PECs for ECs in groundwater. Only four studies were found to estimate the likelihood of an EC reaching groundwater or calculate PECs which all used slightly different approaches. Approaches that only use physico-chemical properties of ECs as leachability indicators can potentially mask or overestimate risks. Nevertheless, it should be acknowledged that these simple approaches can be useful as first steps in the development of monitoring programmes.[52] However, from the studies reviewed it is not clear that they are treated as such, due to the lack of sensitivity testing of results, verification with monitoring data or other prioritisation studies and no mechanisms to update the methods with new data and understanding as they become available.

Depending on physico-chemical properties of the EC does not reflect real environmental conditions.[84,90,91] In theory the two studies which incorporate the vulnerability of groundwater to contamination from ECs should provide a more accurate representation of the risk of exposure. However, similar to the approaches that use only physico-chemical properties of ECs, these methods that incorporate the pathway still do not verify the prioritised results with monitoring data. There is an inherent difficulty in doing this because the fate and transport of ECs may vary in different environmental conditions and there is also a lag time to consider for groundwater due to varying contaminant velocities through in the unsaturated zone. Also, care needs to be taken in the application of tools developed for certain organic contaminants to other ECs, such as acidic pharmaceuticals which may have very different mobility. Therefore, it is not possible to determine a completely unified approach for determining exposure of ECs in groundwater but it is clear that there is a requirement to incorporate new data and research on the sorption and degradation of ECs into any prioritisation approach to improve predictions of exposure.

5 Approaches for hazard assessment of ECs in groundwater

This section provides a review of the methods used to characterise the hazard in each of the studies included (Table 1). Twenty-five of the studies dealt with aquatic ecology as the receptor and sixteen for human health. The approaches for assessing the hazard were grouped into two different types: firstly those that used dose-response data and secondly that used classification data in the scoring system approach.

5.1 Dose-response data hazard assessments

Of the studies that used dose-response data (n = 29), 20 used ecotoxicological data for three trophic levels. Only two studies used mammalian toxicology data. Most of the studies reported using experimental data from existing databases and literature and eight studies reported using QuantitativeStructure-Activity-Relationship (QSAR) data, which estimate the effects based on structural properties of chemical compounds.[108]

Ten studies used human dosage information as indicators of toxicity in humans, applying either the Acceptable Daily Dose (ADI) (n = 6) as "*a measure of the amount of a specific substance in drinking water that can be ingested daily over a lifetime without an appreciable health risk*"[109] or Therapeutic Dose TD (n = 4) as the amount required to have the desired therapeutic effect. Only one study[66], applied the therapeutic dose as a surrogate for toxicity data for aquatic ecology.

For the nine studies that incorporated groundwater, six used dose-response data. Human health was the main receptor considered in these studies (n = 7) and only two considered aquatic ecology. The bioavailability of an EC was generally not accounted for in these approaches, with the exception of one study[10] in this review which corrected the MEC for bioavailability. The lack of experimental ecotoxicological data is considered the norm rather than the exception for many compounds.[10,59] Recent studies have highlighted that ECs require further toxicological data to be developed.[10,55]

Several authors[10,64,72] emphasised that chronic toxicology data sets are the most appropriate to use for hazard assessments of ECs because the main concern relates to long-term exposure at relatively low concentrations. Availability of data for chronic exposure remains low and therefore a reliance on acute data was also highlighted by the same authors.[10,64,61] A conservative approach proposed was to use the lowest available PNEC, even if it is an acute endpoint.[10] Certain health effects cannot be predicted using acute or chronic dose-response tests.[10] In one study[57], different toxicological endpoints for ECs known to have estrogenic activity were used instead. This was the only study where this approach was undertaken, but few details were provided.

5.2 Classification data hazard assessments

Eleven studies used classification data to characterise the hazard. Only two of these studies did not use any dose-repose toxicity data in addition to the classification data. A number of studies incorporated specific long-term health effects data for carcinogenicity (n = 9), mutagenicity (n = 9), teratogenicity (n = 7), endocrine disruption (n = 6) and neurotoxicity (n = 3). Several studies also used persistence (n = 6) and bioaccumulation (n = 11) properties of the EC for prioritising the hazard.

Due the focus on human health in groundwater studies (7 of the 9 studies), the long-term health effects classification approach was used in five of the studies. Only two studies that incorporated groundwater used persistence or bioaccumulation as part of the hazard assessment. The advantages and disadvantages of these approaches for hazard assessment in groundwater are discussed in the following sections.

5.2.1 Classification based on the persistence, bioaccumulation, and toxicity (PBT) assessment

It has been suggested that the reason persistence has often been disregarded in prioritisation approaches is that it is less relevant when there is a continuous discharge into rivers. [57,77] However, for groundwaters persistence is an important factor because more persistent ECs are likely to leach and accumulate in groundwaters. The widespread detection of atrazine in groundwater today, several decades after it ceased being used, is an example of the importance of chemical persistence in groundwater.[28] The PBT assessment is considered useful for circumstances where the risks are difficult to quantify[44], which makes it relevant to the groundwater context. The PBT approach is also used in the United Kingdom to determine if substances are defined as hazardous in groundwater under the WFD and GWDD.[110]

For the assessment of persistence, REACH guidelines[87] definitions of the vP and P was used in the EU WFD prioritisation studies[58-60]. The USA study[57] used a higher threshold of >180 to indicate persistent chemicals. The BIOWIN programme for organic substances can be used to estimate the biodegradability in environmental conditions.[59,72] This was used in the EU WFD prioritisation studies[5860] and a Spanish study[72]. For the assessment of toxicity in the PBT approach the studies generally used the classification under the REACH guidelines[87] or dose-response data, sometimes alongside the risk ratio approach.

For the assessment of bioaccumulation, European guidance recommends that the bioconcentration factor (BCF) for aquatic species is used, mostly from fish.[59,87] The BCF is the ratio of a substance's concentration in an organism and its quantity freely dissolved in ambient water.[59] This approach was used in only three of the studies in this review, with differing thresholds for risk. The $\log K_{ow}$ is also used to estimate a contaminants potential to bioaccumulate within an organism.[64] Two studies[59,64] used a threshold of a $\log K_{ow} > 4.5$ to indicate a risk of bioaccumulation. This threshold originates from EMA guidelines[66,87] which required pharmaceuticals to be screened for further assessment. The USA study[57] used a similar threshold of $\log K_{ow} > 5$, and another study[69] used a threshold of 3, to indicate bioaccumulation potential. One of the studies[64] did highlight the weaknesses of using $\log K_{ow}$ as an indicator of bioaccumulation for pharmaceuticals as they are mostly polar and ionisable. For ECs such as pharmaceuticals there has been little research on their bioaccumulation potential in biota.[111]

5.2.2 Classification based on long-term health effects

Studies that used long-term health effects data for hazard assessment did so to assess the risk to human health. One study[52] scored ECs based on their potential to cause carcinogenic, teratogenic, mutagenic, endocrine disruption and neurotoxic effects. Its authors suggested that this method is more appropriate due to chronic exposure and endpoints such as carcinogenicity and endocrine disruption being realistic hazards. Another advantage of this method is that MECs or PECs are not necessarily required. Another study[69] prioritised the hazard using seven categories for human health effects, which incorporated doseresponse ecotoxicology data for PPCPs and endocrine disrupting compounds for human health.

Both of these studies[52,69] used intermediate scores when there was no data to ensure that they were deemed higher risk than an EC classified as having no effect. It was emphasised by one of the studies[69], that an important issue with these prioritisation approaches was lack of data for many of the health effects categories. The study found that 62% of data in the carcinogenicity category and 82% in the fertility impairment were missing, which resulted in a high uncertainty of results. The lack of an official definition of endocrine disrupting compounds also makes scoring ECs based on this criterion inherently difficult.[60]

Weightings used to assign importance to different criteria are subjective.[71] It therefore is a complex task and the easiest option can be to assign equal weightings to all different categories.[69,72] For example, a study[69] gave equal weight to health effects categories, whereas others such as[, 52,63], weighted the scores to give more importance to carcinogenicity and mutagenicity. Expert judgement is used to assign weightings and can allow decision makers to set priorities; even the importance of different receptors, i.e. human health or aquatic ecology.[69,71]

5.3 Outlook for approaches for hazard assessment of ECs in groundwater

The review highlighted that the use of dose-response toxicity data to characterise the hazard of ECs was the most common approach and only two studies did not use it. However, there is a paucity of toxicity data for many ECs and data is often not accessible due to protection from ‘commercial-in-confidence’.[42]

The main concern of ECs in groundwater relates to long-term exposure at relatively low concentrations. In this context there are some issues with the prioritisation approaches reviewed. In particular the reliance on acute toxicological data rather than chronic toxicological data could misrepresent the risk. Only few studies considered chronic exposure endpoints such as carcinogenicity and endocrine disruption (approximately 28%) but there was a higher portion of the groundwater studies (55%) that did. There are also significant gaps in this type of toxicological classification data which can create high uncertainty in the hazard assessment results.

Unlike surface waters the main source of ECs is not through rapid continuous discharges and therefore the accumulation of ECs is an important consideration. Only two groundwater studies used persistence or bioaccumulation as part of the hazard assessment. There is no standard approach for the assessment of bioaccumulation, and for ECs such as pharmaceuticals the evidence in the literature on their bioaccumulation potential in biota is still limited but is a growing research area.[112-114]

When using the classification approach based on long-term health effects, weightings are generally used to assign importance to a criterion. These weightings are subjective and therefore sensitivity testing should be built into any prioritisation approach to understand the uncertainties and the robustness of the results.

It is clear that greater accessibility and generation of toxicity data for ECs is required and that there are many uncertainties in the hazard assessment approaches. Future approaches for assessment of the hazard of ECs in groundwater should incorporate flexibility to update prioritisation results as new data becomes available, and research on the most appropriated approaches for groundwater are determined and refined.

6 Comparison of prioritisation approaches

Two subsets of prioritisation studies (ECs and pharmaceuticals) were analysed, to compare the chemicals on the prioritised lists to provide an indication of the impact of using different approaches on the results. Only substances that were classified as ECs by the NORMAN network (and not ‘classical’ contaminants) were included in the analysis. The selection process for studies and substances included are described in Supplementary Information A.

The first subset of studies included five studies that prioritised ECs (see Supplementary Information D). There were 37 ECs included in this analysis. Only three ECs were prioritised in more than one of the studies: diazinon, triclosan and estrone. Diazinon is a pesticide and is regulated in some countries so is not typically considered an emerging contaminant.

The second subset of studies included six studies that prioritised human pharmaceuticals (see Supplementary Information D). There were 64 pharmaceuticals included in this analysis. The studies had been carried out in several different countries: United Kingdom[80], two from France[73,64], Iraq[78], Lebanon[81] and China[11]. Three pharmaceuticals were prioritised in five of the six studies: carbamazepine, diclofenac and ibuprofen. Three others were prioritised in four studies: amoxicillin, ciprofloxacin and clarithromycin. Overall a total of 20 of the 64 pharmaceuticals were prioritised in more than one study (31%).

The comparison of prioritisation results for the two subsets of studies, has shown that there were more similarities between the prioritisation studies for pharmaceuticals. This can be attributed to the similarities between the initial lists of chemicals. The initial lists of pharmaceutical studies were generated from similar sources such as prescription and usage data but in several different countries. All the studies used the PEC and dose-response data. One study[73] was an exception, its authors used the MECs and it did have fewer pharmaceuticals in common with the other studies.

Three of the pharmaceutical studies also compared their ranking outcomes to results from other publications. In the first it was found that carbamazepine and ibuprofen were the most prioritised pharmaceuticals among the eight studies they examined.[81] They also highlighted that six (out of 26) of their prioritised pharmaceuticals were not prioritised elsewhere.[81] The second[78] found that amoxicillin, which they ranked highest, also ranked highly in earlier studies in the United Kingdom and Korea ([58,64]). The third study[11] compared its priority list to a previous review of prioritisation results for pharmaceuticals.[114] Nine of the high priority pharmaceuticals had been identified in the previous review.[11] The study's authors also examined previous prioritisation research in France[64] and Switzerland[116] and found that there were similarities to their own list, despite the use of different methodologies.[11]

The two EC studies that had more than one prioritised EC (triclosan and estrone) in common employed quite different methodologies. The initial lists of ECs were generated by different means and from different sources. The studies also had significantly different numbers of substances on their initial lists ranging from 34 to 2024 ECs (including classical chemicals prior to filtering), and were carried out in Europe and the USA. It is not surprising that the prioritisation results from EC studies reviewed here are variable. These studies can include a complex mixture of types of contaminants with different sources and pathways to the aqueous environment. The substances (Table F of the Supplementary Information) included pesticides applied to agricultural land that reaches water via runoff or infiltration through the soil (such as cyanazine or diazinon), pharmaceuticals and industrial fragrances (such as galoxalide) which probably enter the environment via wastewater, as well as plasticisers (bisphenol A) and flame-retardants (perfluorooctanoic acid).

One of the other studies[57] in this review applied two hazard assessment approaches for ECs in surface water: a scoring system that incorporated persistence and bioaccumulation scores (PBT approach); and the risk ratio. The ECs identified by each approach were markedly different. The risk ratio approach yielded the lowest number (n = 41) compared with the PBT approach which used the risk ratio for toxicity (n = 60). Nearly half of the ECs identified by the first approach had relatively low half-lives (<60 days).

It is apparent that comparing the prioritisation results to determine any commonalities is unsatisfactory without first analysing the initial lists (which unfortunately have rarely been provided) and assessing the substance type. Although most studies are therefore not directly comparable, it has been helpful to provide an indication of the parallels between the priority lists for pharmaceuticals.[81] Given the uncertainties with developing any prioritisation approach,[61] it is useful to make an attempt to evaluate the results through comparison with other similar prioritisation approaches. This has been undertaken in some studies (e.g. [11,78,81]), but there was still a lack of analysis of the initial lists and differences between prioritisation methodologies. There is also merit in carrying out more than one prioritisation using different methods, or sensitivity testing results to understand the uncertainty surrounding the prioritised lists. This would minimise the risk of prioritising ECs that are lower risk or missing ones that could be higher risk in the groundwater environment e.g. persistent chemicals.

7 Framework for prioritisation approaches and future outlook

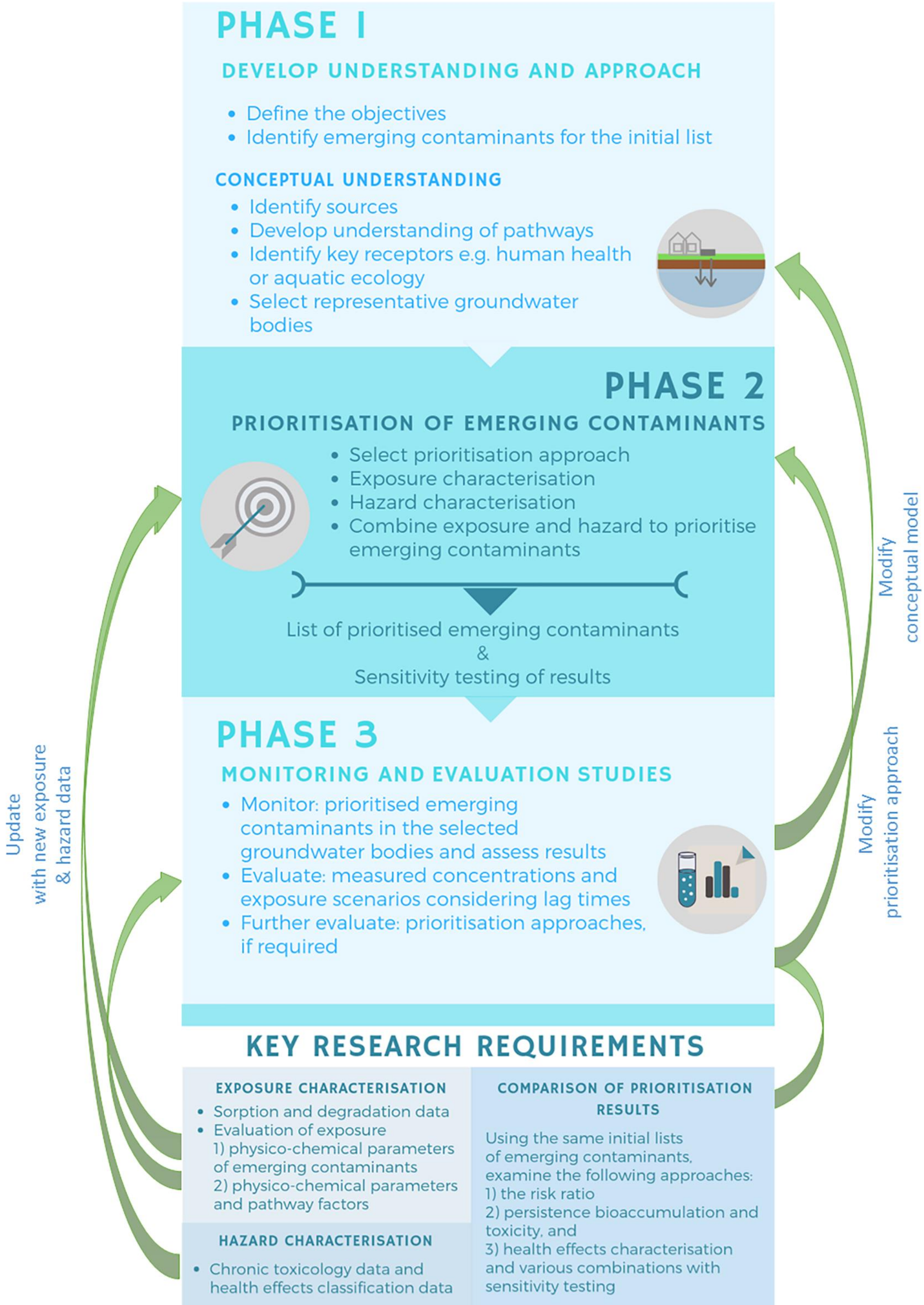
This review has demonstrated that a common approach for prioritising ECs in groundwater has not been developed and verified. Two main issues were revealed by the review. Firstly, the groundwater exposure tools and models examined in this study all had merit, however they need to be confirmed using actual groundwater quality data[50], whilst still considering the lag times. The level of detail required to provide realistic estimates of loading or concentrations in groundwater therefore remains unknown. For example, soil organic carbon was found to be important for sorption of ECs (e.g. [50,54]) but some of the simpler approaches depend on the physico-chemical properties of the EC which often do not reflect real environmental conditions.[84,90,91] There will always be trade-offs between complexity, accuracy and data requirements.[117] The use of MECs is the preferred method for surface water and groundwater when adequate data is available. It is a more reliable method for representing environmental exposure especially for groundwaters where it is difficult to predict concentrations. Careful consideration is required when summarising MEC data and dealing with results below the LOD so that the data is representative of the risk of exposure.

Secondly, the review has highlighted the paucity of toxicity data and physico-chemical data for ECs and issues with access to available data. Due to the risk of ECs accumulating and the potential chronic effects, the long-term health effects classification data will be important to incorporate.[52,81] However, there are still significant gaps in chronic dose-response toxicological data and long-term health effects classification data for ECs.

Lastly this review has indicated that no prioritisation approach is perfect, and it has not demonstrated that one approach is superior to any other, but has highlighted some important advantages and disadvantages. There is an indication that the most comprehensive prioritisation approaches are ones that use a combination of approaches, for example, the risk ratio and scoring methods for hazard assessment (e.g. [57,81]). It has also been shown that the persistence of the EC is important in the groundwater context with the example of carbamazepine accumulation in soil and leaching to groundwater.[104] The dose-response hazard classification can omit highly persistent ECs[57] and the possibility of bioaccumulation of ECs.

The uncertainties with the results of any prioritisation approach require greater effort in scrutinising the results, sensitivity test them and comparing them with other similar studies. Further research is required to analyse the advantages of different prioritisation approaches to optimise the best one for ECs in groundwater. Prioritisation of ECs should not be a static process and improvements in the approaches should be sought and incorporated. Future prioritisation approaches should incorporate flexibility to update prioritisation results as new data becomes available.

Therefore, a broad framework is proposed, that facilitates the incorporation of research on the occurrence, hazards and prioritisation of ECs and an evaluation process. A phased approach adapted for groundwater from Maruya *et al.*[75] is proposed as outlined in Figure 3. The figure also highlights the key priority research areas of: 1) sorption and degradation of ECs in the environment; 2) evaluation of different exposure characterisation approaches to confirm the level of detail required to provide estimates of loading or concentrations in groundwater; 3) the chronic toxicity of ECs and health classification data; 4) comparison of prioritisations approaches for groundwater. For example, the effect of combining prioritisation approaches such as the risk ratio and scoring approaches could be researched, as well as the influence of including persistence and bioaccumulation as factors.



669 **Figure 3 A phased framework for prioritisation of ECs in groundwater incorporating key**
670 **research requirements**

This framework and future research will hopefully enable the prioritisation methodologies to be improved by feeding back results from the evaluations of the prioritisation approach and allowing the incorporation of new data. This phased approach could also be verified with existing monitoring data for ECs in groundwater.

The first phase involves developing the conceptual understanding and approach. The first step is to define the objectives of the prioritisation approach. This ought to be done by both scientists and policy makers or decision makers,[2] determining the priorities (human health, aquatic ecology or both). In addition, it would be important to consider the scale of occurrence to be considered. The WFD, for example, would require that both human health and ecology are considered and the occurrence to be examined at a groundwater body scale.[42,43]

The next step would be to develop the initial conceptual model for the source, pathway and receptors of ECs in groundwater. Given the wide variety of ECs and difference in source types, it is considered that a pragmatic approach would be to develop scenarios that focus on certain sources and groups of substances that can then be tested.[42,75] Such scenarios for groundwater could be designed around the current understanding of the sources of ECs (see Figure 1[17]). One exemplary scenario could relate to the ECs found in wastewater discharges from septic tanks and other private treatments works which discharge to groundwater. It would also be important to monitor in areas with high and low groundwater vulnerability to be representative of the risk spectrum[43] and allow different groundwater vulnerability settings to be tested.

The second phase involves the actual prioritisation process. A prioritisation approach that is appropriate for groundwater and the identified receptors needs to be selected. An initial priority list of ECs to be monitored can be developed based on the characterisation of environmental exposure and hazard. The results of the prioritisation of ECs should be sensitivity tested to understand the level of uncertainly around the data used and any scores or weightings. Carrying out more than one prioritisation approach using the same initial list and a further analysis of the resulting priority lists would help to ensure that results are robust and that the decision makers are informed of the uncertainties that require consideration.

The third phase involves monitoring the prioritised ECs in groundwater and verifying the conceptual models and exposure characterisation by relating the MECs to the exposure scenario.[75] The monitoring can then be adapted as needed based on the monitoring results and evaluation studies (see Figure 3).[75] The lag times between environmental release and occurrence in groundwater needs to be reflected. This evaluation step can also be used to adapt the prioritisation approach if required and reassess prioritisation results. Additional feedback loops are proposed to incorporate new physico-chemical data and hazard classification and toxicity data.

The framework addresses the problem of the lack of knowledge on occurrence and fate of ECs, and uncertainties surrounding prioritisation results. The prioritisation process needs to be dynamic and responsive as new information becomes available, for example, through the proposed voluntary European GWWL process, refinements can also be made to the conceptual models and subsequent priority lists. The framework will ultimately enable further groundwater monitoring data to be gathered for ECs that pose the highest risk to groundwater receptors, while paving the way for an optimised approach for prioritising ECs in groundwater.

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9 Supporting Information

The Supporting Information (SI) available includes: Systematic review search protocol (A); Search strings used for database and web searches (B); Study characteristics from systematic review (C); and results tables of results from comparison of prioritization studies (D).

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